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**Effectiveness of artificial floods for benthic community recovery after sediment flushing from  
a dam**

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15    **Abstract**

16

17    The number of dams is predicted to increase worldwide under the current global change scenario. A  
18    major environmental problem associated with dams is the release of large quantities of fine sediment  
19    downstream. Therefore, future studies in river conservation will largely be focused on the  
20    management of sediments trapped by reservoirs. The aim of this study was to investigate the  
21    downstream ecological impacts of sediment flushing from a dam and the effectiveness of artificial  
22    flash floods as a recovery strategy. Artificial flash floods have often been employed to remove large  
23    amounts of sediment from riverbeds, but their importance in improving the biological quality of lotic  
24    environments is almost unknown. We carried out a series of quantitative macroinvertebrate samplings  
25    over a 2-year period that started before sediment release and included the artificial flushing events.  
26    We characterized the macroinvertebrate community in its structural and functional aspects, and tested  
27    the performance of two biomonitoring indexes, comparing their diagnostic ability. Our results  
28    demonstrated that sediment flushing significantly altered the structure and composition of benthic  
29    communities for more than 1 year. Flash floods exacerbated the overall biological quality, but we  
30    believe that this treatment was useful because, by removing large amounts of sediment, the biological  
31    recovery process was accelerated. Finally, regarding the water quality assessment, we found that the  
32    biomonitoring index for siltation, composed of a selection of taxonomical and functional metrics, was  
33    more reliable than the generic one.

34

35

36    **Keywords:** macroinvertebrates, siltation, dam, multi-metric index, Alpine stream, restoration.

37

## 38 Introduction

39

40 Dam and reservoir management is likely to represent one of the biggest challenges in river  
41 conservation over the next years. Dams are increasing both in number and in geographical extent  
42 (Fox et al. 2016), but numerous aspects of their correct management remain unresolved. In some  
43 countries there is a growing tendency to remove large dams (Gangloff 2013; Grant and Lewis 2015),  
44 while the current and anticipated changes in hydrological regimes with climate change are causing  
45 an increase in dam construction worldwide (O'Connor et al. 2015; Zarfl et al. 2015).

46 Dams can have a significant ecological impact on lotic ecosystems, disrupting natural flow regimes  
47 and altering naturally evolved processes (Van Cappellen and Maavara 2016; Arheimer et al. 2017).  
48 For example, dams inhibit natural dispersal pathways and movements of aquatic organisms, causing  
49 population isolation and extirpation of migratory fauna (Olden 2016). Dams transform upstream lotic  
50 habitats into lentic environments, with a strong impact on structural and functional composition of  
51 aquatic coenoses (Gray and Ward 1982; Guareschi et al. 2014; Rothenberger et al. 2017), and  
52 interrupt the natural downstream transport of sediments. Moreover, the quality of water released from  
53 reservoirs is often significantly altered during reservoir residence (WCD 2000). Dams generally trap  
54 all the bedload, i.e. gravel and coarse sand particles, and much of the suspended load, which has  
55 detrimental effects on many natural processes, both in downstream and impounded reaches (Brandt  
56 2000). Downstream sections are usually sediment starved causing substrate armoring and increased  
57 bank erosion and riverbed incision (Kondolf 1997; Graf 2006), with many detrimental ecological  
58 effects. By contrast, in the upstream impounded reach, sediment trapping in reservoirs dramatically  
59 alters the morphological and ecological features of the systems, decreasing substrate particle size and  
60 transforming the river into a lentic body (Tang et al. 2016; Rothenberger et al. 2017). Moreover,  
61 sediment accumulation within impoundments reduces storage capacity of reservoirs, interfering with  
62 the functional purpose of the dam and reducing the efficiency of associated hydroelectric power plants  
63 (Annandale 2013).

64 Unfortunately, loss of storage due to sedimentation was rarely considered in the design of many dams,  
65 resulting in a widespread problem of reservoir sediment management (Morris and Fan 1998; Owens  
66 et al. 2015), with an estimated 28% of the global sediment flux being potentially trapped in reservoirs  
67 (Vorosmarty et al. 2003). There are three main strategies for managing sedimentation in reservoirs.  
68 The first - sediment yield reduction - aims to reduce sediment inflow to reservoirs by controlling  
69 erosion in the upstream river network (Quiñonero-Rubio et al. 2016). The second - sediment routing  
70 - a modern approach whereby inflow sediments are allowed to pass through or around the reservoir

(Sumi and Kantoush 2010; Auel and Boes 2011, Martín et al. 2017). Unfortunately, these approaches are not suitable for many existing dams. The last, most common strategy is sediment removal by eliminating accumulated sediment through excavation, dredging, or drawdown flushing (Espa et al. 2013; Kondolf et al. 2014).

Sediment flushing can restore or preserve reservoir storage but often has dramatic negative impacts on freshwater ecosystems. In some cases, sediment flushing causes the release of an excessive quantity of fine material, forming thick layers of sediments, which can settle in the river for years before being transported downstream by the flow, affecting the diversity and abundance of the river biota. The adverse effects associated with the unnatural deposition of fine sediment in the river bed have been documented for different aquatic groups, including macroinvertebrates (Wood and Armitage 1997; Mebane 2001; Bo et al. 2007; Bilotta and Brazier 2008; Jones et al. 2012), fish (Kemp et al. 2011; Sear et al. 2016) and plants (Izagirre et al. 2009; Jones et al. 2014). To accelerate removal of these deposits, dam managers can implement artificial flash floods, i.e. sudden releases of water from the dam for washing away and moving the settled sediments downstream. Artificial floods have been applied for achieving restoration goals, for example, to reproduce the natural flow regime in heavily regulated streams affected by multiple dams in their basins (Robinson et al. 2003, 2004, 2012).

Given the complexity and unpredictability of the flushing operations, data on their impact and effectiveness are scarce. Several operational strategies have been proposed and applied in the last years for successfully desilting reservoirs; for example, it is recommended to perform flushing operations during the period of maximum seasonal run-off to maintain the natural pattern of sediment transport (White 2001; Kondolf et al. 2014). Moreover, flushing-related sedimentation can be mitigated by providing additional water from nearby reservoirs, and stopping sediment release at night to limit the chronic stress associated with these activities (Espa et al. 2013; Quadroni et al. 2016). However, the effects of sediment flushing on aquatic communities are often underestimated due to the lack of pre-impact data and specific biomonitoring tools. We therefore aimed to analyze the impact of sediment flushing from an Alpine river dam on downstream benthic macroinvertebrate communities in terms of richness, density and structure; and to monitor the temporal recovery of these coenoses after two artificial flash floods. Finally, we compared the diagnostic power of two biomonitoring indexes based on macroinvertebrates, one generic and the other designed for detecting siltation effects.

## **Materials and methods**

104

## 105 *Study Area*

106 The study was carried out in the Stura di Demonte river (hereinafter referred to as Stura), which is  
107 considered one of the least impacted rivers in the entire Italian Alpine area. In fact, the Stura hosts  
108 populations of endangered fish species, such as marble trout (*Salmo marmoratus*), grayling  
109 (*Thymallus thymallus*) and European bullhead (*Cottus gobio*). For this reason, the SCI (Sites of  
110 Community Importance) Stura di Demonte (Code IT1160036) was established according to the EU  
111 43/1992 Directive (“Habitat Directive”) to protect the most conserved river stretch, which includes  
112 four municipalities between the villages of Demonte and Roccasparvera. The study area was located  
113 downstream of the Roccasparvera Dam (44.340256°, 7.440539°; Fig. 1), which was built in 1957 and  
114 subsequently acquired by the Italian power company, Enel Green Power, in March 1963.  
115 Roccasparvera is a 13 m concrete overflow dam, with a slightly arched planimetric course and a total  
116 capacity of  $0.580 \times 10^6 \text{ m}^3$ . The land use of the watershed is dominated by forests (48%), followed  
117 by pastures (39%) and rocky areas (10.8%), while the urbanized and agricultural areas account for  
118 only for 1.4% and 0.8%, respectively.

119

## 120 *Sedimentation and dam operations*

121 Due to a safety-related emergency, in January 2016, a sudden and rapid depletion in the volume of  
122 water in the reservoir was necessary. On 7<sup>th</sup> January at 10:15 AM, the sluice gates on the dam were  
123 opened, allowing the water to flush at  $5.4 \text{ m}^3 \text{ s}^{-1}$ ; the discharge was progressively increased to  $13.7$   
124  $\text{m}^3 \text{ s}^{-1}$  (at 02:45 PM), and was successively reduced until the reservoir was empty at the end of the  
125 flushing event on 8<sup>th</sup> January 2016 at 02:00 AM ( $4.5 \text{ m}^3 \text{ s}^{-1}$ ). During flushing, water turbidity,  
126 suspended solids concentration (SSC), water temperature, dissolved oxygen (DO), pH, and electric  
127 conductivity were monitored continually by Enel Green Power at the station ROC1 (44.338820°,  
128 7.448782°), located 0.7 km downstream. This event produced a significant environmental impact,  
129 with a massive layer of sediments being accumulated for almost 8 km downstream of the dam  
130 (personal observations). These sediments filled the river bed of the Stura and numerous irrigation  
131 canals, completely transforming their morphology. Enel Green Power and the local environmental  
132 authority (Province of Cuneo) agreed on a recovery strategy aimed at accelerating the removal of  
133 sediment from the impacted reach. Since such a large amount of sediment would have taken many  
134 years to disperse with normal flows, sudden water releases (artificial flash floods) were implemented  
135 on two occasions (18<sup>th</sup> and 25<sup>th</sup> May 2016; Table 1), which increased erosion and transport in the

136 downstream section. In both circumstances, they started at 10:30 AM and consisted of a progressive  
137 increase of released water every 5 minutes: from  $10 \text{ m}^3 \text{ s}^{-1}$  to  $41 \text{ m}^3 \text{ s}^{-1}$ . This latter condition was kept  
138 constant until 01:00 PM and then the volume of sluiced water was progressively reduced.

139

#### 140 *Sampling design*

141 Benthic macroinvertebrate samples were collected in ten surveys from December 2015 (i.e., pre-  
142 impact) to December 2017 (Table 1). Ten quantitative samples were collected randomly in each  
143 survey, using a Surber sampler (250  $\mu\text{m}$  mesh size;  $0.062 \text{ m}^2$  area). Organisms were counted in the  
144 laboratory and identified to the family level using taxonomic keys (Campaioli et al. 1994; 1999).

145

#### 146 *Statistical analysis*

147 Statistical analysis was performed using R (R Development Core Team 2017). Changes in the  
148 composition and structure of benthic invertebrate communities throughout the study were visually  
149 evaluated using Principal Coordinate Analysis (PCoA). A Bray-Curtis similarity matrix was used for  
150 multivariate ordination of the samples, while a Permutational Analysis of Variance (PERMANOVA)  
151 was applied to test significant effects of the “sampling session” on the composition and structure of  
152 benthic invertebrate communities. These analyses were carried out with the functions “vegdist”,  
153 “betadisper” and “adonis” of the R package *vegan* (Oksanen et al. 2015). Univariate analysis was  
154 then performed to detect the specific impacts of the sediment release and the temporal recovery of  
155 macroinvertebrate communities. Four different community metrics were considered, namely total  
156 taxa richness, total density of macroinvertebrates ( $\text{Ind m}^{-2}$ ), EPT (Ephemeroptera, Plecoptera and  
157 Trichoptera) richness and EPT density. Each Surber sample was considered as a replicate, and prior  
158 to performing regression models outliers were removed using the method of Zuur et al. (2010) for  
159 data exploration. Generalized Additive Models (GAMs) were employed to test the non-linear  
160 response of the selected community compared to time in days from the sediment release; for this  
161 operation, negative values were assigned to the pre-impact data (9<sup>th</sup> December 2015). All the GAMs  
162 were carried out using the “gam” function of the *mgcv* R package (Wood and Wood 2015) and  
163 applying a Poisson distribution. A negative binomial distribution was then applied in case of  
164 overdispersion.

165 Finally, our data were analyzed in the context of river quality assessment, comparing the performance  
166 of two selected indexes, namely the STAR\_ICMi and the multi-metric index proposed by Doretto et  
167 al. (2018). The former was originally developed as a calibration tool in the context of the Water

168 Framework Directive (EU 60/2000 Directive), and is currently the official biomonitoring index  
169 provided by Italian regulations to classify the ecological status of running waters. STAR\_ICMi is a  
170 multi-metric index composed of six community metrics, belonging to three categories (i.e. diversity,  
171 abundance, and sensitivity/tolerance). The weighted values of these metrics are aggregated in the  
172 STAR\_ICMi index, and the final score is obtained by the ratio between the observed and the reference  
173 value (for details see Bo et al. 2017). The latter instead represents a recently proposed multi-metric  
174 index using benthic invertebrates designed to detect the impacts of river siltation. This index includes  
175 two diversity metrics (i.e., total taxa and EPT richness) and a functional metric (i.e., the abundance  
176 of taxa preferring shallow, fast-flowing water with coarse substrates: Ecological Group A *sensu*  
177 Usseglio-Polatera et al. 2000). Compared to the STAR\_ICMi, which is generic, the multi-metric  
178 index is a stressor-specific tool designed to assess the effects of siltation (for further details see  
179 Doretto et al. 2018). These indexes were calculated for each sampling occasion, pooling together the  
180 ten macroinvertebrate samples collected on that date. The temporal trends of the selected indexes  
181 were thereby compared to evaluate which method provided the best performance in relation to the  
182 sediment release.

183

## 184 **Results**

185

### 186 *Sediment and water chemistry parameters*

187 The flushing operations dramatically altered the physical and chemical parameters of the water, with  
188 the greatest changes occurring during the night between 7<sup>th</sup> and 8<sup>th</sup> January 2016 (Fig. 2). An  
189 exponential increase in SSC was documented; the SSC varied from 0.00 before the flushing to 10 mg  
190 L<sup>-1</sup> at the end of the operation (mean = 12.07 mg L<sup>-1</sup>), with intermediate peaks. The highest level of  
191 SSC occurred on 8<sup>th</sup> January from 00:00 AM to 10:00 AM; during this period the SSC was  
192 consistently higher than 20 mg L<sup>-1</sup>, reaching a peak of 31.05 mg L<sup>-1</sup> (Fig. 2a). A very similar trend  
193 was also observed for the turbidity, as this parameter was highly correlated with the SSC. From the  
194 onset to the end of the flushing operations, the water turbidity increased from 0.80 to 2160 NTU,  
195 reaching a maximum value of 6209 NTU (mean = 2413 NTU). During the most intense phase of  
196 flushing, the turbidity was higher than 4300 NTU (Fig. 2b). Water temperature varied from 3.80 to  
197 5.28 °C (mean = 4.26°C) but a sharp decline was recorded between 7:15 PM and 09:15 PM, when  
198 the water temperature dropped to 0.26 °C, but then abruptly increased again (Fig. 2c) possibly due to  
199 a thermal stratification of the impounded reach. The DO concentration markedly reduced during the



monitored period from 12.50 mg L<sup>-1</sup> at the beginning of the operations to 5.01 mg L<sup>-1</sup> at 02:00 AM (Fig. 2d). DO then slowly increased again, reaching a value of 7.09 mg L<sup>-1</sup> at the end of the operations (mean = 8.90 mg L<sup>-1</sup>). A similar temporal trend was also observed for the pH (Fig. 2e), which varied from 8.20 to 7.14 (mean = 7.70). Around 01:00 AM, the pH dropped to 7.17 and then remained relatively constant until the end of the flushing activity. Finally, marked oscillations were observed in relation to the electric conductivity over the monitored period (Fig. 2f). This parameter varied from 447  $\mu$ S cm<sup>-1</sup> at the beginning of the operations to 483  $\mu$ S cm<sup>-1</sup> at the end (mean = 421  $\mu$ S cm<sup>-1</sup>). However, the lowest and highest values were 225 and 502  $\mu$ S cm<sup>-1</sup>, respectively.

### *Benthic invertebrates*

We collected 7,860 organisms belonging to 38 families from nine different systematic groups. Diptera and Trichoptera were the orders with the highest number of families, nine and eight, respectively, whereas Gastropoda, Crustacea and Triclada were each represented by only one family. The number of taxa per sample ranged from 1 to 21 (mean = 9), while the total density of benthic invertebrates ranged from 40 to 8440 Ind m<sup>-2</sup> (mean = 1638 Ind m<sup>-2</sup>). Mayflies - Heptageniidae and Baetidae - as well as dipteran larvae of the families Chironomidae and Simuliidae were the most abundant taxa.

PCoA ordination of the samples depicted a clear temporal shift in the composition and structure of benthic invertebrate communities (Fig. 3) and PERMANOVA results showed a significant effect of the sampling occasion ( $F_{9,86} = 9.9$ ;  $P < 0.001$ ). Compared to the pre-impact situation (i.e. 2015.12.09), the communities just after the sediment release were significantly different in terms of composition and structure, and oriented at the right side of the plot. The macroinvertebrate communities slowly approached the pre-impact community levels only after both of the flash floods (from 2016.07.22 onward), reaching full overlap with the pre-flushing community on the last sampling occasion (i.e. 2017.12.12, nearly 2 years later).

Results of the multivariate analysis were confirmed and further strengthened by those of the GAMs (Table 2). The sediment release and consequent flash floods significantly reduced the total taxa richness, which showed a unimodal pattern with a negative peak around 150 days after the sediment release (Fig. 4a). Taxa richness then progressively recovered over time, reaching values comparable to pre-impact levels after 600 days. The combined effects of sedimentation and flash floods also resulted in a similar temporal pattern for the EPT richness (Fig. 4b). Prior to impact, the average number of EPT families was ten, which subsequently dropped to three after the sedimentation event and to one after the flash floods. Again, the results of the model showed a progressive increase in the

232 EPT richness over time, with an almost complete recovery after 600 days. Conversely, GAMs  
233 illustrated very fluctuating patterns in the density of benthic invertebrates. Both the total (Fig. 4c) and  
234 the EPT density (Fig. 4d) showed a dramatic depletion after the sediment release as well as the flash  
235 floods. However, compared to the taxa and EPT richness, both these two events were followed by  
236 peaks in density. A less pronounced decrease was also observed after 400 days, and then both the  
237 total and EPT density increased again to nearly pre-impact values. It is interesting to note that these  
238 marked temporal fluctuations were mainly driven by the extremely high abundance of Baetidae (Fig.  
239 5a), Chironomidae (Fig. 5b) and Simuliidae (Fig. 5c) specimens during the first post-release sampling  
240 occasions (Table 2). Thus, the specific response of these few taxa accounted for the observed results  
241 for the total and EPT density.

242

#### 243 *Biomonitoring indexes*

244 The two biomonitoring indexes for assessing the impact of the sediment release from the  
245 Roccapervera dam gave different responses (Fig. 6). The STAR\_IMCi (Fig. 6a) on the pre-impact  
246 sampling occasion was 1.1 and then decreased as a consequence of the sedimentation event and flash  
247 floods. However, these changes only corresponded to a slight deterioration in the water quality  
248 classification, which shifted from High to Good and only dropped to Scarce on the sampling occasion  
249 2016.06.06 (i.e., after the second flash flood, STAR\_ICMi = 0.25). After the artificial floods, the  
250 index increased, ranging from 0.80 to 1.15, and it reached the High quality class on the last two  
251 sampling occasions (i.e. after 475 days). Overall, the STAR\_ICMi detected a change in the water  
252 quality class, but this only resulted in a shift from High to Good for most of the recovery period, with  
253 the exception of the sampling occasion 2016.06.06. Conversely, the temporal variation of the multi-  
254 metric index for siltation proposed by Doretto et al. (2018) depicted a different temporal pattern (Fig.  
255 6b). From the pre-impact value of 0.83, this index dropped markedly after the sediment release (0.33)  
256 and the flash floods, achieving a minimum value of 0 on the sampling occasion 2016.06.06. The index  
257 then increased over time, reaching full recovery on the last sample date. Compared to the  
258 STAR\_ICMi, this stressor-specific index showed a more pronounced variation between the pre-  
259 impact value and that recorded just after the sediment release. Moreover, the temporal pattern of this  
260 index was more similar to the community metrics, especially taxa richness.

261

#### 262 **Discussion**

263 Although several studies have documented the effects of sediment releases from dams, long-term  
264 series encompassing pre-impact data and multi-year sampling campaigns, like this study, are rare.  
265 Usually, flushing operations are programmed over long periods of time, and carried out by alternating  
266 sediment and water pulses between the day-time and night-time, or by providing volumes of diluting  
267 water from adjacent reservoirs (Crosa et al. 2010; Espa et al. 2013; 2015). These practices allow  
268 minimization of the ecological impacts of excessive releases of sediments and facilitate the recovery  
269 of aquatic communities.

270 Unfortunately, adopting these approaches is not always possible according to the specific features of  
271 the dam and the magnitude of the related impacts. The sediment flushing from the Roccasparvera  
272 dam, documented in this study, was concentrated in a short period (from 7<sup>th</sup> January at 10:15 AM to  
273 8<sup>th</sup> January at 02:00 AM). In just a few hours, drastic alterations of all the main physical and chemical  
274 attributes of water were detected; for example, the SSC and turbidity were in the range of three orders  
275 of magnitude greater than before the sediment release. At the same time, a dramatic reduction in the  
276 concentration of DO and pH was observed. Similar changes have been documented by other authors  
277 in relation to flushing events (Gray and Ward 1982; Peter et al. 2014; Khakzad and Elfimov 2015,  
278 Espa et al. 2016) and have led to a strong impact on benthic invertebrate assemblages. We found a  
279 temporal shift in the composition of communities; pre-impact macroinvertebrate communities were  
280 mainly composed of EPT taxa, while after the sediment release and flash floods, the number of taxa  
281 belonging to these orders was significantly reduced. Our results corroborate other studies carried out  
282 in the Alpine area (Espa et al. 2013; Peter et al. 2014; Martín et al. 2017). More profound effects were  
283 evident when macroinvertebrate density was considered, with a collapse of the total and EPT density  
284 due to the sediment release, which was exacerbated with the flash floods. Similar results were also  
285 observed by other authors (Doeg and Khoen 1994; Quadroni et al. 2016); for example, Crosa et al.  
286 (2010) studied the ecological consequences of a reservoir flushing in an Italian alpine stream. They  
287 found the density of invertebrates dropped, on average, from 2000 to 100 Ind m<sup>-2</sup>. In our study, 2  
288 years were necessary to achieve full recovery of the benthic communities to their pre-impact taxa  
289 richness and density of macroinvertebrates. However, taxa richness recovered slowly and  
290 progressively, while there was a fluctuating response of density. The extremely high abundance and  
291 dominance of few generalist taxa, such as Baetidae, Chironomidae and Simuliidae, may account for  
292 the temporal pattern observed for density. Several authors have demonstrated that these families are  
293 early colonists and tend to be dominant during the first phases of post-disturbance recolonization,  
294 including experimental floods from reservoirs (Otermin et al. 2002; Robinson et al. 2004). Compared  
295 to our results, a faster post-flushing recovery of macroinvertebrate communities has been documented  
296 in other studies (Crosa et al. 2010; Espa et al. 2013; 2016). This delayed recovery pattern could be

297 explained by the fact that the flash floods were performed 5 months after the sedimentation event.  
298 However, we postulate that, in the long-term, these artificial treatments contribute to restoring the  
299 natural habitat for benthic invertebrates, facilitating the recovery of communities.

300 Another possible factor that could account for the full community recovery is the overall pristine  
301 conditions of the investigated stream, which was reflected by the excellent pre-impact ecological  
302 status of the sampled reach. As expected, the sedimentation process produced a deterioration of the  
303 water quality, but this mainly consisted in a shift from High to Good, even when the river bottom was  
304 buried by a large amount of fine sediment (personal observation) and the density and richness of  
305 benthic invertebrates were highly reduced. The water quality further dropped to Scarce as an  
306 immediate consequence of the flash floods. Although these operations were successful in removing  
307 the fine sediment and promoting long-term recovery, they initially induced stressful conditions for  
308 benthic invertebrates, dislodging them from the substrate. This may account for the low water quality  
309 on that occasion, after which the water quality quickly re-established to Good and then High. In  
310 particular, despite the temporary worsening of the biological quality after the flash floods, we believe  
311 that this treatment had a considerable importance overall, because large amounts of sediment were  
312 removed, which would have taken much longer to disperse naturally, thus the biological recovery  
313 process was accelerated. The limited performance of STAR\_ICMi was not completely unexpected  
314 since it is a generic multi-metric index, as confirmed by other authors who obtained similar results  
315 when assessing the impacts of sedimentation from dams (Espa et al. 2015; 2016). On the contrary,  
316 the performance of the index for siltation recently proposed by Doretto et al. (2018) offered  
317 interesting results, although no ecological classification has been provided yet. Nevertheless, when  
318 looking at the temporal trend of the index, it was more reliable than the STAR\_ICMi in detecting the  
319 effects of disturbance in relation to both the sedimentation release and the flash floods. Quadroni et  
320 al. (2016) evaluated the consequences of sediment flushing in an Alpine stream and stated that the  
321 STAR\_ICMi can underestimate impacts, resulting in an imprecise water quality assessment. This is  
322 the first time that our stressor-specific index has been tested using an independent dataset. Overall,  
323 its performance offered promising applications and confirmed the useful role of macroinvertebrates  
324 as bioindicators for running waters (Rosenberg and Resh 1993; Buss et al. 2015). Moreover, the  
325 combination of taxonomical and functional metrics considered here appears very successful for  
326 detecting the specific effects of siltation, and this supports the adoption of functional groups for river  
327 biomonitoring (Merritt et al. 2015).

328 While several solutions have been proposed in relation to the timing of the sluiced water and the  
329 amount of flushed fine sediment, such as thresholds in the SSC and discharge, the development of  
330 stressor-specific methods based on the biota response still remains unexplored (Crosa et al. 2010;

331 Quadroni et al. 2016). Our report therefore offers the application of a diagnostic biomonitoring tool  
332 designed to evaluate the specific impacts associated with siltation. Since the number of reservoirs and  
333 dams is predicted to increase in the near future (Fox et al. 2016; Van Cappellen and Maavara 2016),  
334 developing science-based biomonitoring and management tools is an important challenge for river  
335 ecologists and managers.

336

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343

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485

486 **Tables**

487

488 **Table 1** Sampling scheme: Date = date, Label = labels used in the statistical analysis  
489 (year.month.day), Activity = data and information associated to each date

Date	Label	Activity
9 <sup>th</sup> December 2015	2015.12.09	Pre-impact data
7 <sup>th</sup> - 8 <sup>th</sup> January 2016	-	Sediment flushing
23 <sup>rd</sup> March 2016	2016.03.23	Benthic invertebrates sampling
17 <sup>th</sup> May 2016	2017.05.17	Benthic invertebrates sampling
18 <sup>th</sup> May 2016	-	First flash flood event
24 <sup>th</sup> May 2016	2016.05.24	Benthic invertebrates sampling
25 <sup>th</sup> May 2016	-	Second flash flood event
06 <sup>th</sup> June 2016	2016.06.06	Benthic invertebrates sampling
22 <sup>nd</sup> July 2016	2016.07.22	Benthic invertebrates sampling
09 <sup>th</sup> December 2016	2016.12.09	Benthic invertebrates sampling
15 <sup>th</sup> March 2017	2017.03.15	Benthic invertebrates sampling
27 <sup>th</sup> April 2017	2017.04.27	Benthic invertebrates sampling
12 <sup>th</sup> December 2017	2017.12.12	Benthic invertebrates sampling

490

491

492 **Table 2** Statistics of the Generalized Additive Models: Est = estimate, SE = standard error, z = z-  
493 value, P = p-value,  $X^2$  = Chi-square

Metric	Intercept				Smooth	
	Est	SE	z	P	$X^2$	P
Taxa richness	2.124	0.037	57.080	< 0.001	146.900	< 0.001
Total density	6.721	0.047	143.900	< 0.001	698.100	< 0.001
EPT richness	1.426	0.054	26.530	< 0.001	122.900	< 0.001
EPT density	5.781	0.047	121.900	< 0.001	893.600	< 0.001
Baetidae density	4.118	0.054	76.180	< 0.001	821	< 0.001
Chironomidae density	4.980	0.048	102.900	< 0.001	915	< 0.001
Simuliidae density	3.667	0.053	68.790	< 0.001	1173	< 0.001

494

495

496 **Figure captions**

497

498 **Fig. 1** Area of study: the Stura di Demonte watershed and the investigated stream reach

499

500 **Fig. 2** Values of: **a** concentration of suspended solids (SSC), **b** turbidity, **c** water temperature, **d**  
501 concentration of dissolved oxygen (DO), **e** pH and **f** conductivity measured at the ROC1 station  
502 during the sedimentation release from the Roccasparvera dam

503

504 **Fig. 3** Principal Coordinate Analysis (PCoA) ordination plots based on the dissimilarity matrices  
505 (Bray-Curtis). Symbols represent the benthic community samples collected in each sampling  
506 occasion (year.month.day). The grey lines link each sample with its corresponding centroid (black  
507 dots)

508

509 **Fig. 4** Representation of the GAM smoother applied to time interacting with the community metrics:  
510 **a** total taxa richness, **b** EPT richness, **c** total density and **d** EPT density. The black line represents the  
511 smoothed function (i.e.  $s(\text{Time}, \text{degrees of freedom})$ ), while the grey area indicates the 95%  
512 confidence interval. Black vertical line indicates the sediment flushing, while the dotted vertical lines  
513 represent the two artificial floods. Grey ticks on the x-axis represent the sampling occasions

514

515 **Fig. 5** Representation of the GAM smoother applied to time interacting with the density of: **a**  
516 Baetidae, **b** Chironomidae and **c** Simuliidae. The black line represents the smoothed function (i.e.  
517  $s(\text{Time}, \text{degrees of freedom})$ ), while the grey area indicates the 95% confidence interval. Black  
518 vertical line indicates the sediment flushing, while the dotted vertical lines represent the two artificial  
519 floods. Grey ticks on the x-axis represent the sampling occasions

520

521 **Fig. 6** Bars indicate the values of **a** the STAR\_ICM index and **b** the sedimentation index (Doretto et  
522 al. 2018) calculated for each sampling occasion (year.month.day)

523